SIMULATING MANAGEMENT EFFECTS ON PHOSPHORUS LOSS FROM FARMING SYSTEMS

D. M. Sedorovich, C. A. Rotz, P. A. Vadas, R. D. Harmel

ABSTRACT. *A process-level soil phosphorus (P) model including surface and subsurface components was incorporated into the Integrated Farm System Model (IFSM). Model evaluation indicated that sediment losses were adequately estimated compared to observed data for a corn production system in Texas. In a further evaluation, sediment losses simulated for a wide range in cropping systems and tillage practices were similar to those predicted by the current state-of-the art erosion estimation model (WEPP). Total P losses were accurately predicted when manure P was applied at suitable rates of less than 250 kg P ha^{-1}, but at higher application rates overestimation of P loss was found. Compared to observed data, soluble P loss was underestimated and sediment P loss was overestimated, but this was primarily due to a difference in the differentiation between soluble and sediment P between the modeling and experimental studies. To illustrate the use of the model, IFSM simulations were performed to evaluate the effects of manure handling and tillage systems on P loss from farms in Pennsylvania. For a 100-cow dairy farm, a manure handling strategy that used a 6-month storage and application by injection decreased total P loss by 19% compared to daily surface application, but annual farm net return was decreased by $57/cow. Compared to conventional tillage with a moldboard plow, use of conservation tillage and no-till systems reduced total P loss by 46% and 57%, respectively, with small increases in farm profitability. Reduced tillage increased soluble P loss, suggesting that conservation and no-till systems should be combined with practices such as manure injection to reduce all forms of P loss. The enhanced IFSM containing the soil P model provides a tool for whole-farm analysis of management effects on P loss along with other environmental and economic considerations.*

Keywords. Farm, Model, Phosphorus loss, Simulation.

The U.S. Environmental Protection Agency estimates that there are 22,000 impaired surface waters (e.g., lakes, streams, and reservoirs) in the country, with 11% of these impairments due to nutrients originating primarily from agriculture (U.S. EPA, 2003a, 2003b). Because phosphorus (P) is a primary contributor to eutrophication in surface waters, nonpoint-source P pollution from agriculture is a major concern in the U.S. (Sharpley et al., 1999).

Research on P management at the farm level is focused on implementing alternative management practices in an attempt to reduce the amount of P lost from the farm, which should decrease the P entering receiving waters. If these management strategies reduce the profitability of the farm, though, the practices are unlikely to be implemented. Thus, strategies to reduce the impact of nonpoint-source P pollution from agriculture must be evaluated considering the economics of the whole farm.

Computer modeling has emerged as a cost-effective and relatively rapid method of analyzing different farm management strategies. One model, the Integrated Farm System Model (IFSM), simulates a whole-farm system including crop growth, livestock performance, economics, and nutrient flows (Rotz and Coiner, 2006). This research tool is used to analyze the long-term effects of implementing various cropping, feeding, manure handling, and other farm management strategies. This farm model simulates the transformations of N during manure handling and the associated volatilization, leaching, and denitrification losses from farms. A weakness of IFSM has been that the P cycle was not simulated. Instead, good P management was assumed on the farm, with P loss through runoff and other processes fixed at 5% of the total P applied to farmland in manure and fertilizer.

Our goal was to create a model that simulates the effects of different management strategies on farm-level P loss. Specific objectives were to make a dynamic, field-scale model of the major soil P processes, evaluate the model against observed field losses of P, incorporate this component into the farm simulation model (IFSM), and demonstrate the usefulness of the improved version of IFSM by simulating the impacts of alternative tillage and manure management practices on P loss.

MODEL DESCRIPTION

The soil P cycle is a complex process consisting of various chemical forms and transformations of P. A process-based

The authors are **Dawn M. Sedorovich, ASABE Member Engineer,** Research Assistant, Department of Agricultural and Biological Engineering, The Pennsylvania State University, University Park, Pennsylvania; **C. Alan Rotz, ASABE Fellow,** Agricultural Engineer, USDA-ARS Pasture Systems and Watershed Management Research Unit, University Park, Pennsylvania; **Peter A. Vadas,** Research Scientist, USDA-ARS U.S. Dairy Forage Research Center, Madison, Wisconsin; and **R. Daren Harmel, ASABE Member Engineer,** Agricultural Engineer, USDA-ARS Grassland Soil and Water Research Laboratory, Temple, Texas. **Corresponding author:** C. Alan Rotz, USDA-ARS, Bldg. 3702, Curtin Rd., University Park, PA 16802; phone: 814-865-2049; fax: 814-863-0935; e-mail: Al.Rotz@ars.usda.gov.

soil P model was developed to simulate soil P dynamics at the field and farm scales. The model uses relationships from the Erosion–Productivity Impact Calculator (EPIC) (Williams, 1995; Jones et al., 1984) and the Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998; Neitsch et al., 2002) with modifications by Vadas et al. (2004, 2005) to better represent surface processes. The major components include surface and soil P pools and the transformations and flows that link these pools. Since many of the relationships used in the soil P model were previously verified and documented, only the major relationships and their integration for this farm–scale application are described here.

For this application, a general soil model is used to predict the moisture and nutrient flows through the soil profile under each crop (Rotz and Coiner, 2006). Precipitation, runoff, evapotranspiration, moisture migration, and drainage are tracked through time to predict the moisture content in multiple layers of the soil profile. Soils are generally described as clay loam, loam, sandy loam, and loamy sand with deep, moderate, or shallow depths. The soil is modeled in four layers, where the top three layers are relatively thin surface layers with thicknesses of 30, 45, and 75 mm. The fourth layer extends from the 150 mm depth to the bottom of the soil profile or the crop rooting depth, whichever is first limiting.

SURFACE P

When simulating livestock farming systems, surface application of manure is an important process. Previous models that simulate the soil P cycle have not adequately represented surface P (Williams, 1995; Arnold et al., 1998). In these models, unincorporated surface applications of manure and fertilizer are added directly to either the organic or inorganic soil P pools, depending on the source. This does not include loss directly from the P source on the surface and therefore underestimates the amount of P lost in runoff. Field data have shown that P loss directly from a surface application of manure can be significant when rain occurs soon after application (Kleinman and Sharpley, 2003). Therefore, the soil P model incorporates a surface P model proposed by Vadas (2006) and Vadas et al. (2005, 2007) for simulating surface application of manure.

Four surface P pools are used to simulate surface applications and soil interactions (Vadas, 2006). These pools represent water–extractable inorganic (MW_{ip}) and organic (MW_{op}) P and non–water–extractable inorganic (MT_{ip}) and organic (MT_{op}) P (fig. 1). Surface processes include surface application, runoff, and transformation along with soil–surface interactions through infiltration and tillage. Additions to the surface P pools occur through surface application of manure. The freely draining portion of applied manure with a high moisture content infiltrates into the soil immediately after application (process 1 in fig. 1). The remaining P is proportioned into the four surface pools based on the application method and characteristics of the applied manure.

After P is added to the surface pools, it is released from both water–extractable pools (MW_{ip} and MW_{op}) during rainfall events. The amount of inorganic P released from the surface pools (P_{rel}, kg P) is a function of the water–to–manure ratio (W, cm^3 water g^{-1} manure dry matter) and the water–extractable inorganic P on the surface (MW_{ip}, kg P) (Vadas et al., 2007):
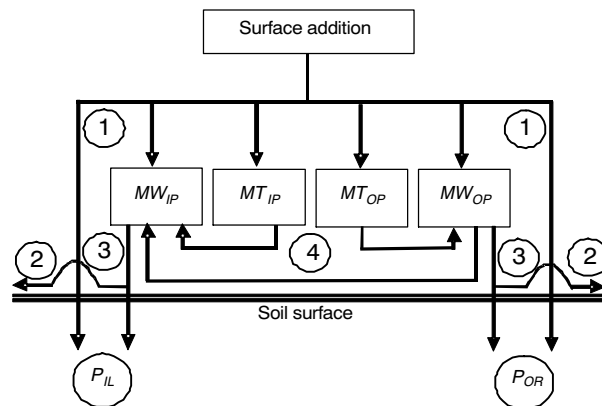

Figure 1. Surface P pools and processes simulated in the soil P model.

$$P_{rel} = 1.2 \left(\frac{W}{W + 73.1} \right) MW_{ip} \qquad (1)$$

The concentration of inorganic P released from the surface pools by rainfall (P_{conc}) is the mass of P released (P_{rel}) divided by the total volume of precipitation (i.e., the product of precipitation depth and land area).

If runoff occurs from a rain event, a portion of the inorganic P concentration enters the runoff water and is lost from the system (process 2 in fig. 1). The P lost (P_{runoff}, kg P ha^{-1}) is the product of the runoff depth (Q, mm), the concentration of inorganic P released from the surface pools (P_{conc}, mg P L^{-1}), and a P distribution factor (P_{dist}, dimensionless factor ranging from 0 to 1):

$$P_{runoff} = \frac{Q \cdot P_{dist} \cdot P_{conc}}{100} \qquad (2)$$

The P distribution factor is empirically modeled as a function of the runoff depth per unit of precipitation depth (Vadas et al., 2007). Runoff is calculated using the USDA Soil Conservation Service (SCS) runoff curve number method. With this method, the amount of runoff is related to the amount of precipitation and the moisture content in the top 45 cm of the soil profile (Rotz and Coiner, 2006).

The remaining water–extractable P, or the total amount released if there is no runoff, infiltrates into the soil and enters the appropriate soil P pools (process 3 in fig. 1). The water–extractable inorganic P is added to the upper soil layer pool of labile P, and the water–extractable organic P is added to the upper layer organic P pool. Infiltration of P following surface application and prior to rainfall is set at 60%, 50%, 20%, and 0% for liquid (<7%, DM), slurry (7% to 12%, DM), semi–solid (12% to 20% DM), and solid manure (>20% DM), respectively (Vadas, 2006).

The final surface process is the decomposition of the non–water–extractable P (MT_{ip} and MT_{op}) into water–extractable P (MW_{ip} and MW_{op}) (process 4 in fig. 1). The mass of P decomposed (P_{decom}, kg P d^{-1}) is the product of a dynamic rate factor ($Rate$, d^{-1}) and the mass of P on the surface (P_{mass}, kg P):

$$P_{decom} = Rate \cdot P_{mass} \qquad (3)$$

The dynamic rate factor is the product of three dimensionless factors representing the effects of ambient temperature, manure moisture content, and the age of the manure on the surface (Vadas et al., 2007). The decomposed organic and in-

organic P is subtracted from the respective surface pools. This decomposed P is added to the water–extractable surface pools, with 25% of the decomposed organic P added to the organic pool and 75% added to the inorganic pool (Vadas et al., 2007).

For subsurface application of manure or fertilizer, the surface pools are essentially bypassed, with inorganic and organic P components added directly to the appropriate soil pools. Subsurface injection of manure is modeled assuming 95% infiltration, which places the remaining 5% of the applied P in surface pools. Subsurface–applied inorganic fertilizer is added to the labile pool of the second soil layer.

INORGANIC SOIL P

The inorganic soil P component of the model is based on relationships from EPIC (Williams, 1995; Jones et al., 1984) and SWAT (Arnold et al., 1998; Neitsch et al., 2002) with modifications to simulate rapid adsorption and desorption as suggested by Vadas et al. (2006). Three inorganic soil P pools are simulated, representing labile (P_{il}), active (P_{ia}), and stable (P_{is}) P (fig. 2). The P_{il} pool is the P in solution and weakly sorbed to soil particles. This labile P is readily desorbed and thus provides the amount available for crop uptake and runoff loss. The P_{ia} pool is the non–labile P in balance with the labile pool. The P_{is} pool is the P that is least susceptible to plant uptake and runoff loss and that which is in balance with the active pool.

Inorganic pool processes include the transfer of P between the labile and active pools, which represents rapid adsorption and desorption. Rapid adsorption maintains the dynamic equilibrium between the labile and active P pools, and rapid desorption represents the opposite process. Similarly, the movement of P between the active and stable pools represents slow adsorption and desorption. Phosphorus is also taken from the labile pool through crop uptake, runoff, and leaching loss.

The rate of P movement (R_{la}, kg P ha^{-1} d^{-1}) from the inorganic labile pool (P_{il}, kg P ha^{-1}) to the inorganic active pool (P_{ia}, kg P ha^{-1}) is a function of a dynamic rate factor (K_{la}, d^{-1}) and the expected P distribution between the two soil reservoirs (P_{Bal}, kg P ha^{-1}), which is a function of a P sorption factor (P_{sp}, dimensionless) (Vadas et al., 2006):

$$R_{la} = K_{la} \cdot P_{bal} \tag{4}$$

where

$$P_{bal} = P_{il} - P_{ia} \left[\frac{P_{sp}}{1 - P_{sp}} \right] \tag{5}$$

The dynamic rate factor is a function of the number of days since an imbalance occurred between the pools, the P sorption factor, and two empirical parameters (Vadas et al., 2006). Slow adsorption and desorption are similarly defined, but
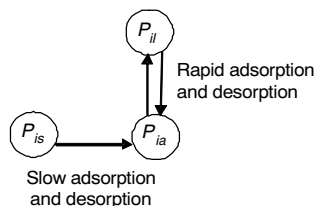

Figure 2. Inorganic soil P pools and processes simulated in the soil P model.

they occur between the active P pool and the stable P pool. The rate of P movement (R_{as}, kg P ha^{-1} day^{-1}) from P_{ia} to the inorganic stable P pool (P_{is}) is a function of a rate constant based on soil characteristics K_{as} (day^{-1}) and P_{is} (kg P ha^{-1}) (Vadas et al., 2006):

$$R_{as} = K_{as} \left(4P_{ia} - P_{is} \right) \tag{6}$$

An important process simulated for the inorganic pools is the loss of P from the upper soil layer through runoff. Using the theory of an extraction coefficient, labile P is withdrawn from the soil reservoir and enters runoff. The mass of soluble P lost in runoff (P_{sol}, kg P ha^{-1}) is a function of the runoff depth (Q, m), the extraction coefficient (C_{extr}, Mg m^{-3}), the soil depth (D_{layer}, m), and the soil bulk density (ρ_{BD}, Mg m^{-3}):

$$P_{sol} = \frac{P_{il}(Q)(C_{extr})}{D_{layer}(\rho_{BD})} \tag{7}$$

This loss of P from the upper soil layer is added to that lost from the surface pools through runoff (eq. 2) to obtain the total soluble P loss.

Models such as EPIC and SWAT use extraction coefficients that are specific to soil type, hydrological conditions, land use, and other conditions. However, Vadas et al. (2005) found a strong relationship between soil P and runoff dissolved P for a variety of soil types and hydrology, which suggests that a single extraction coefficient can be used to simulate the loss of P in runoff under various conditions. Thus, a single extraction coefficient (C_{extr} = 0.005 Mg m^{-3}) is used in the soil P model.

A portion of the soil P is removed through crop uptake. Uptake is a function of the difference between the optimal P concentration for a given crop and the actual concentration in the simulated plant material (Jones et al., 1984). The P uptake of the crop on each simulated day is subtracted from the labile P pools within the soil profile. Uptake is weighted to draw primarily from the upper three soil layers where most of the soil P is located.

Leaching loss of soil P through the soil profile is normally relatively small and unimportant, but it can occur. A relatively simple relationship is used to predict this loss based on the work of Vadas (2001). The total soil P leached from the root zone of the crop on any given day is the sum of that leached from the top soil (top three soil layers) and subsoil (bottom layer). Soil P leached from each of these layers (P_{lch}, kg ha^{-1} d^{-1}) is a function of the P concentration in the leachate (C_{lp}, mg kg^{-1}), the amount of leachate occurring on that day (L_s, mm d^{-1}), the depth of the soil layer (D_{layer}, m), and the depth of the root zone in the soil profile (D_{soil}, m):

$$P_{lch} = 0.01(C_{lp})(L_s) \left[\frac{D_{layer}}{D_{soil}} \right] \tag{8}$$

The concentration of P in the leachate is exponentially related to the inorganic labile P in the soil layer (P_{il}) (Vadas, 2001):

$$C_{lp} = e^{(P_{il} - EQ_i)/EQ_s} \tag{9}$$

where EQ_i is the intercept and EQ_s is the slope of a logarithmic relationship between sorbed P and disolved P. The values of EQ_i and EQ_s are determined using empirical relationships
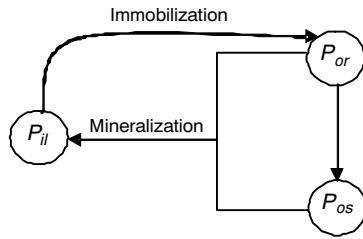
Figure 3. Organic soil P pools and processes simulated in the soil P model.

with soil clay content (m_c) derived from the data of Vadas (2001):

$$EQ_s = 1.49\,(m_c) + 6.18 \qquad (10)$$

$$EQ_i = 4.89\,(EQ_s) - 6.51 \qquad (11)$$

The amount of leachate is predicted on a daily time step based on soil moisture content and other soil characteristics (Rotz and Coiner, 2006).

ORGANIC SOIL P

The organic soil P component is based on published equations from EPIC (Williams, 1995; Jones et al., 1984) and SWAT (Arnold et al., 1998; Neitsch et al., 2002) with no further modification. Two P pools are simulated: residue organic P (P_{or}) and stable organic P (P_{os}) (fig. 3). The P_{or} pool represents organic P in residue and microbial biomass; P_{os} represents organic P in a more stable, less available form.

Organic soil transformations consist of mineralization and immobilization. Mineralization involves the net conversion of organic P (both P_{or} and P_{os}) to inorganic labile P (P_{il}). Immobilization is the reverse process, with P moving from P_{il} to P_{or}. The two organic P pools also interact, with a fraction of the organic P in crop residue (P_{or}) becoming less available and moving to the stable P pool (P_{os}).

The net mineralization from both organic P pools (R_p, kg P ha^{-1} d^{-1}) is a function of the rate of mineralization of P from decaying organic matter (R_{pr}, kg P ha^{-1} d^{-1}), the rate of P mineralization from stable organic matter (R_{pos}, kg P ha^{-1} d^{-1}), and the rate of P immobilization by decomposing organic matter (R_{upr}, kg P ha^{-1} d^{-1}):

$$R_p = 0.8 R_{pr} + R_{pos} - R_{upr} \qquad (12)$$

As documented by Jones et al. (1984), R_{pr} is a function of a rate constant for decomposition of decaying organic matter, moisture and temperature constants, and the C:N and C:P ratios; R_{pos} is a function of a rate constant for decomposition of stable organic matter and the moisture and temperature constants; and R_{upr} is a function of the rate of organic matter decomposition and the microbial P concentration.

SEDIMENT P AND EROSION

Sediment P loss is simulated using enrichment ratios to predict bioavailable and labile P loss as a function of erosion sediment loss (Sharpley, 1985). On any day when erosion occurs, enrichment ratios are determined for both bioavailable and labile P as exponential functions of the amount of sediment loss that occurs that day (Sharpley, 1985). Bioavailable P loss in sediment is the product of the sediment loss, the bioavailable P concentration in the upper soil layer, and the enrichment ratio. Bioavailable P is the sum of the active and

stable inorganic and organic pools in the upper soil layer. Similarly, labile P loss is the product of the sediment loss, the inorganic labile P concentration in the upper soil layer, and the labile P enrichment ratio. The sum of the bioavailable and labile P losses provides a total sediment P loss.

Erosion sediment loss is predicted using the Modified Universal Soil Loss Equation (MUSLE). Sediment loss (*sed*, kg ha^{-1}) is a function of the daily runoff depth (Q, mm), the peak runoff rate (q_{peak}, m^3 s^{-1}), the area analyzed (A_{hru}, m^2), and five factors representing soil erodibility (K), slope length (L), slope steepness (S), cover management (C), and a support practice (Ps) (Neitsch et al., 2002):

$$sed = 11.8 \left(Q \cdot q_{peak} \cdot A_{hru} \right)^{0.56} \cdot K \cdot L \cdot S \cdot C \cdot Ps \qquad (13)$$

The soil erodibility factor is determined using relationships published by Williams (1995), where K is the product of four dimensionless empirical factors: f_{c-sand}, a factor that gives low values for soils with a high percentage of coarse sand; $f_{clay-silt}$, a factor that gives low values for soils with high clay to silt ratios; f_{orgC}, a factor that gives low values for soils with high organic carbon content; and $f_{hi-sand}$, a factor that gives low values for soils with very high sand content:

$$K = f_{c-sand} \left(f_{clay-silt} \right) \left(f_{orgC} \right) \left(f_{hi-sand} \right) \qquad (14)$$

These four factors are functions of the sand, silt, clay, and organic carbon contents of the surface soil, as documented by Williams (1995) and Neitsch et al. (2002). The method published by Renard et al. (1996) is used to calculate L, S, and C for the conditions of each field. Since IFSM does not have a mechanism to simulate support practices, the support practice factor Ps was set to a default value of one.

TILLAGE OPERATIONS

Tillage operations can have a major effect on both sediment and soluble P losses. Tillage loosens the soil surface, which may increase sediment and sediment–bound P losses. Tillage also incorporates the surface layer of P and mixes the upper soil layers. This incorporation and mixing reduces the potential for soluble P loss in runoff.

The major tillage effect is the incorporation and mixing of the surface and upper three soil layers. On the day in which a tillage operation occurs, soil P is mixed among the layers using a tillage efficiency factor. Assigned tillage efficiencies are 100%, 40%, 5%, and 90% for moldboard plow, chisel plow, no–till planting, and manure injection operations. This efficiency defines the portion of each P component in a given layer that is mixed with the other layers. For example, in a chisel plow operation, 60% of each P component in each layer remains in that layer, while 40% is mixed and uniformly redistributed among the surface and upper three soil layers.

Tillage also affects the cover management factor (C, eq. 13) in the prediction of sediment loss. Factors affecting C include prior land use and surface roughness (Neitsch et al., 2002). The prior land use factor is 1.0 on the day tillage occurs and declines through time toward a minimum value of 0.45. To represent no–till systems, this factor is held at this minimum value. The surface roughness factor is a function of a roughness index with values of 1.9, 1.2, and 0.4 assigned for moldboard plow, chisel plow, and no–till tillage systems, respectively.

INTEGRATION WITH FARM MODEL

The soil P model was specifically developed for use in the whole-farm simulation model IFSM. This farm model simulates the many biological and physical processes of a crop, beef, or dairy farm (Rotz and Coiner, 2006). Crop production, feed use, and the return of manure nutrients back to the land are simulated over many years of daily weather. Each year is simulated starting from the same initial conditions without carryover of feeds, manure, or other inventories. Initial soil moisture and nutrient levels are reset to the same user-defined values at the beginning of each simulated year. Therefore, the model really simulates the annual production of a system for 25 independent years of weather.

Growth and development of alfalfa, grass, corn, soybean, and small grain crops are simulated on a daily time-step based on soil and weather conditions. Tillage, planting, harvest, and storage operations are simulated to predict resource use, timeliness of operations, crop losses, and nutritive changes in feeds. Feed allocation and animal response are related to the nutritive value of available feeds and the nutrient requirements of the animal groups making up the herd.

Nutrient flows through the farm are modeled to predict potential nutrient accumulation in soil and loss to the environment (Rotz and Coiner, 2006). The quantity and nutrient content of the manure produced is a function of the quantity and nutrient content of the feeds consumed. Nitrogen volatilization occurs in the barn, during storage, following field application, and during grazing as influenced by weather and manure management practices. Denitrification and leaching losses from the soil are related to the rate of moisture movement and drainage from the soil profile, as influenced by soil properties, rainfall, and the amount and timing of manure and fertilizer applications. A whole-farm balance of N, P, and K includes the import of nutrients in feed, fertilizer, deposition, and legume fixation and their export in crops, milk, animals, manure, and losses leaving the farm.

The total P excreted by animals on the farm is that consumed in feed minus that used in growth and milk production, and the P concentration is the total excreted P divided by total manure DM (Rotz and Coiner, 2006). The fraction of total P in the water-soluble inorganic form is a function of the P content in animal diets. From Dou et al. (2002), readily soluble P (P_s, g kg^{-1} fecal DM) is a function of dietary P concentration ($DietP$, g kg^{-1} feed DM):

$$P_s = 1.37 \, DietP - 2.8 \tag{15}$$

where P_s is limited to a minimum value of 1.5 g kg^{-1}. P_s is averaged over the intake of all mature animals on the farm. This fraction is divided by the fecal DM concentration of total P to get the fraction of total P in the water-extractable inorganic pool.

Simulated farm performance measures include crop yields and quality, harvest and storage losses, feed use, animal production, manure quantity and nutrient content, and the labor, energy, machinery, and other resources used each simulated year. Based on this performance, production costs, income, and the net return to management (i.e., profit) of the farm are determined for each year (Rotz and Coiner, 2006). By modeling several alternatives, the relative effects of system changes on resource use, production efficiency, environmental impact, and net return are compared. The distribution of annual values obtained in a given simulation can be used to assess risk over variable weather conditions.

The new soil P component model was integrated with the whole-farm model by linking inputs to other farm components. Major inputs from other farm components include the quantity and P content of the manure produced each year, the timing of manure application, and the type and timing of tillage operations. Other information obtained from the farm model includes soil characteristics, land topography, daily weather information, and daily runoff from each crop area.

MODEL EVALUATION

The soil P model was evaluated to determine its adequacy in estimating sediment loss as well as soluble, sediment, and total P loss at the field scale. This evaluation included a comparison of IFSM-simulated losses to observed data and error analyses of the model. Both the soil and surface relationships of the soil P model have been extensively evaluated in prior works (Jones et al., 1984; Sharpley et al., 1984; Vadas et al., 2004, 2005, 2007, 2006). Therefore, our evaluation was focused on the prediction of field-level losses and management effects on those losses.

Because the soil P model was designed to predict losses at the field or farm scale, model evaluation required observed data at this scale. Full-year data were not available for fields with dairy manure application that included adequate documentation of the weather and management information required for simulation. Therefore, a field-scale study was used that included poultry litter fertilization on corn fields near Riesel, Texas (Harmel et al., 2004). Although the conditions of this study were very different from those used in model development and calibration, this evaluation was used because: (1) this was the most suitable data available, and (2) this provided a more robust evaluation of the model.

In the Texas study, six small watersheds with similar Houston Black clay soils were monitored over three years (Harmel et al., 2004). Because uniform production practices were used throughout each watershed, they represented field-scale losses. The study consisted of one fallow year (2001) and two cultivated years (2002-2003) with tillage, planting, harvest, and manure application dates similar for each field (table 1). Five of these watersheds received litter applications (Y8, Y10, Y13, W12, and W13), and one served as the control (Y6). IFSM input files for weather, farm management, and machinery were created to replicate the conditions of the six watersheds during 2002 and 2003. Observed and simulated data for runoff, sediment erosion loss, and each form of P loss (soluble, sediment, and total P) were compared using a regression analysis (table 2).

RUNOFF AND EROSION

The model was able to simulate the total annual runoff occurring from each of the watersheds with good accuracy (table 2). Simulated runoff was highly correlated to observed values with a slope near one and a low intercept. This was primarily due to the calibration of the model for this specific evaluation procedure. The curve number used to control runoff in the model was adjusted to provide similar runoff to that observed so that differences in runoff would not confound the more important comparisons in this particular model evaluation, which were sediment and P losses in runoff.

Table 1. Selected manure and management characteristics of six small watersheds in corn and wheat production that were fertilized with different amounts of poultry litter (Harmel et al., 2004).

Characteristic		Watershed					
		Y6	Y8	Y10	Y13	W12	W13
Area (ha)		6.6	8.4	7.5	4.6	4.0	4.6
Slope (%)		3.2	2.2	1.9	2.3	2	1.1
Curve number		87	87	87	87	87	87
Litter rate (Mg ha^{-1} year^{-1})		0.0	13.4	6.7	4.5	9.0	11.2
Mean N rate (kg ha^{-1} year^{-1})		168	370	278	237	296	328
Mean P rate (kg ha^{-1} year^{-1})		19	358	196	122	229	286
Mehlich P (mg kg^{-1})	2002	20.9	51.7	40.9	43.5	55.1	68.3
	2003	17.7	91.2	63.9	45.0	62.6	111.2
Land use/crop	2002	Corn	Corn	Corn	Corn	Corn	Corn
	2003	Corn/wheat[a]	Corn/wheat	Corn/wheat	Corn/wheat	Corn/wheat	Corn/wheat
Curve number	2002	85	83	90	85	83	85
	2003	89	85	89	87	84	85

[a] Double-cropped with corn followed by winter wheat.

Table 2. Observed and IFSM-simulated soluble, sediment and total P losses, erosion, and runoff for the study watersheds in 2002 and 2003 (Harmel et al., 2004).

Watershed	Year	P Application Rate (kg ha^{-1})	Soluble P (kg P ha^{-1})		Sediment P (kg P ha^{-1})		Total P (kg P ha^{-1})		Erosion (kg ha^{-1})		Runoff (mm)	
			Obs.	IFSM	Obs.	IFSM	Obs.	IFSM	Obs.	IFSM	Obs.	IFSM
Y6	2002	0	0.252	0.173	1.573	0.530	1.825	0.703	1619	2121	206.1	202.6
	2003	0	0.260	0.128	0.491	0.398	0.751	0.525	677	1832	148.1	149.0
Y8	2002	434	1.327	5.583	1.766	0.873	3.093	6.455	1455	1265	179.9	171.9
	2003	250	0.802	1.610	0.427	0.603	1.229	2.213	503	935	104.5	104.7
Y10	2002	259	1.858	6.303	1.089	0.758	2.947	7.060	1337	2040	296.5	304.7
	2003	125	1.099	1.445	0.258	0.433	1.357	1.878	482	1164	157.3	148.9
Y13	2002	159	0.813	2.843	2.246	0.813	3.058	3.655	1948	1849	229.4	238.3
	2003	89	0.387	0.618	0.436	0.378	0.822	0.995	512	978	107.7	105.0
W12	2002	304	0.649	3.700	2.073	0.700	2.722	4.400	1807	1171	162.6	171.9
	2003	117	0.671	0.755	0.561	0.385	1.232	1.140	726	798	97.5	96.3
W13	2002	370	1.695	5.515	1.717	0.685	3.412	6.200	1605	939	194.9	202.6
	2003	178	1.186	1.320	0.405	0.443	1.591	1.763	461	586	105.7	105.0
Mean		--	0.917	2.499	1.087	0.583	2.003	3.082	1094	1307	166	167
Std. Dev.		--	0.529	2.247	0.742	0.179	0.976	2.398	583	520	60	63
r^2			0.70		0.65		0.80[a]		0.26		0.99	
Slope			3.56		0.19		2.19		0.45		1.05	
Intercept			-0.77		0.37		-1.31		810		-7.0	

[a] A regression of the eight points with P application rates of 250 kg ha^{-1} or less gave a slope of 1.1, intercept of 0, and correlation of 0.61.
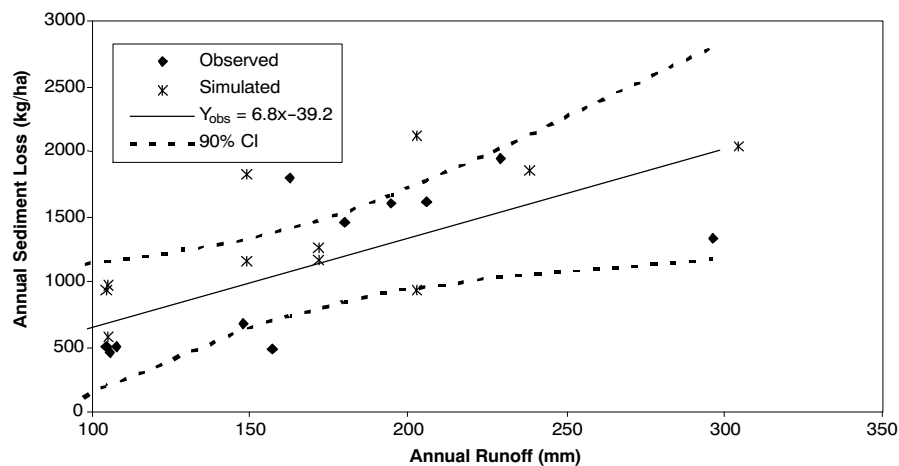

Figure 4. Relationship between erosion losses and runoff for IFSM-simulated and observed data for six small field-size watersheds near Riesel, Texas (2002 and 2003).

Simulated and observed erosion sediment loss values were similar in magnitude, but not highly correlated with one another (table 2). As a further evaluation, simulated and observed sediment losses were plotted as a function of runoff (fig. 4). A 90% confidence interval around the regression curve of the observed data illustrated the unexplained variation. Although the variance of data was large, the observed data showed a general increase in sediment loss of 7 kg ha^{-1} per mm increase in runoff. Simulated losses followed this same trend, with most individual data points falling within the 90% confidence interval of the observed data. For the simulated values that fell outside the confidence interval, there were similar observed values outside the confidence interval.

An analysis was performed to further evaluate the accuracy of the erosion component of IFSM when predicting sediment losses across a wide range of crop and tillage practices. This was done by comparing IFSM–simulated sediment losses to values predicted by the Water Erosion Prediction Project (WEPP) model (Flanagan and Nearing, 1995), a widely accepted model for erosion prediction. Input files were created for IFSM and WEPP for seven scenarios in crop and tillage management. These scenarios were corn established using conventional, conservation, and no–till practices; soybeans established using these three tillage practices; and established alfalfa. Each scenario was evaluated over nine years of weather from Madison, South Dakota, using each model. Model parameters were set to represent actual conditions at this location using a medium clay loam soil (60.8% silt, 32.2% clay, 7% sand, and 3.7% organic matter) with a soil slope length of 22.1 m and slope angle of 5.6%.

The two error measures used to compare IFSM and WEPP simulated values were the percentage difference and the root mean square error (RMSE). The difference between WEPP and IFSM simulated soil losses for individual crop and tillage practices ranged from –56% to 25%, with no effect from crop use (table 3). Neither the percentage difference nor the RMSE indicated a consistent difference between the predictions of the two models. A comparison across this wide range of practices showed a high correlation between the losses predicted by the two models (fig. 5). Losses simulated with IFSM were higher than those predicted by WEPP when conventional tillage was used. The largest error occurred with corn production using no tillage; all other points showed good agreement between the two models. Based on these results, IFSM is a good indicator of expected sediment loss that predicts as well as the current state–of–the–art model WEPP.

Table 3. Comparison of WEPP and IFSM average annual sediment loss estimations for different crop and tillage practices based on simulations for nine years of weather at Madison, South Dakota.

Crop and Tillage[a]	Avg. Annual Sediment Loss (kg ha^{-1} year^{-1})		Diff. (%)	RMSE
	WEPP	IFSM		
Corn, conventional	3292	4097	25	3819
Corn, conservation	993	986	–1	683
Corn, no-till	688	304	–56	490
Soybean, conventional	4911	5504	12	4150
Soybean, conservation	2771	2786	1	2265
Soybean, no-till	1951	1523	–22	1207
Established alfalfa	334	244	–27	352

[a] Tillage operations represent conventional (moldboard plow), conservation (chisel plow), and no–till establishment.
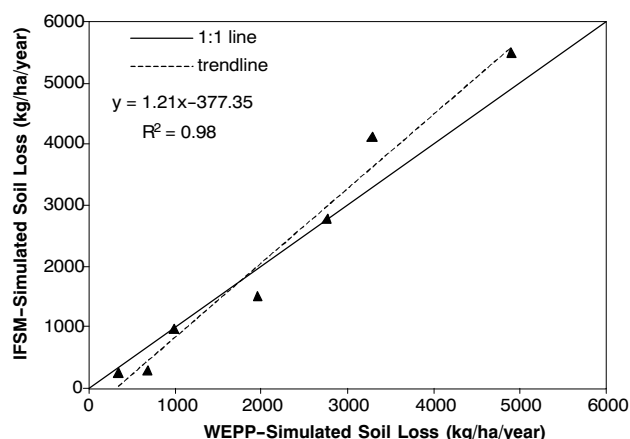

Figure 5. Comparison of WEPP and IFSM simulated annual sediment losses over a wide range in crop and tillage practices based on simulations for nine years of weather at Madison, South Dakota.

PHOSPHORUS LOSS

Simulated losses of soluble and sediment P were correlated with observed values over the two years for the six watersheds near Riesel, Texas (table 2). The slopes of simulated versus observed values showed that simulated soluble P losses were considerably greater than observed values, particularly at high values of loss. The opposite occurred with sediment P losses, where simulated losses were less than predicted, particularly at higher values of loss. Total P losses were well correlated with observed data (r^2 = 0.8), but again simulated losses tended to be greater than observed at higher values of loss. In 2001 during a fallow production season, total P losses were measured in the Riesel study of similar magnitude as those simulated for the 2002 and 2003 corn production years; however, lower losses were observed during the last two production years (Harmel et al., 2004).

For further evaluation, simulated and observed data were compared as a function of manure application rate (fig. 6). The observed data showed a small increase in total P loss of about 50 g ha^{-1} for each kg P ha^{-1} increase in applied manure. Simulated losses were similar to those observed for application rates of 250 kg P ha^{-1} or less. At higher application rates, though, simulated losses were substantially greater than those observed. Application rates greater than 250 kg P ha^{-1} would normally be considered excessive. Phosphorus applied at this level is considerably greater than typical crop requirements or crop removal rates. Since the soil P model was developed to represent conditions on well managed farms, a better evaluation of the model is to compare losses at application rates of 250 kg P ha^{-1} or less. Over this range, the model did very well at simulating total P losses. A regression of observed and simulated losses for application rates of 0 to 250 kg P ha^{-1} (lines 1, 2, 4, 6, 7, 8, 10, and 12 of table 2) had a slope of 1.1 with an intercept near zero and a correlation of 0.61.

The proportions of soluble and sediment P in the total P losses were substantially different between simulated and observed values (table 2). Averaged over all six watersheds for 2002 and 2003, simulated soluble P was more than twice that observed, and sediment P was about half that observed. This difference was primarily due to a difference in the definition of sediment P. In the model, sediment P was defined as that attached to sediment as determined using an enrichment ratio
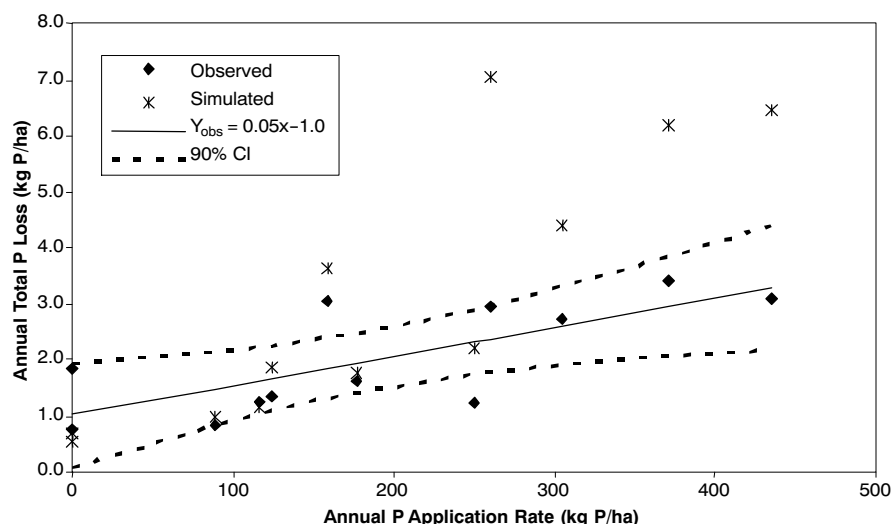
Figure 6. Relationship between total P losses and P application rate for observed and IFSM–simulated data for six field–size watersheds near Riesel, Texas (2002 and 2003).

(Sharpley, 1985). In the field study, sediment P was measured as that which settled from solution over a 3 to 5 day period (Harmel et al., 2004). This settling procedure likely included organic manure P carried in the runoff that was not attached to sediment. Therefore, this procedure would measure more sediment and less soluble P than that simulated. This difference between simulated and observed values was again less when the P application rate was 250 kg P ha^{-1} or less (table 2). Considering this difference in definition, soluble and sediment P losses were adequately simulated as they were defined in the model.

MODEL APPLICATION

To evaluate the usefulness of the soil P component model in estimating P loss from farms, five manure handling strategies and three tillage options were simulated for a representa-

tive dairy farm in Pennsylvania using IFSM. The farm included 100 Holstein cows and 85 replacement heifers on 90 ha of medium–depth, clay–loam soil with an initial labile P concentration of 38 ppm. Crops produced included alfalfa and grass, primarily harvested as silage; corn harvested as silage and high–moisture grain; and oats harvested as high–moisture grain and straw bedding. Lactating cows were fed a total mixed ration in confinement housing, while older heifers and dry cows were on pasture during the grazing season. Farm performance was simulated with 25 years of weather for south–central Pennsylvania.

This simulation analysis was included to demonstrate the use of the model in estimating soil P loss as affected by farm management, not to provide a comprehensive comparison of specific production systems. Thus, documentation of model parameters is limited to the major differences among the management options simulated. The facilities and equipment used in the simulations are listed in table 4 along with as–

Table 4. Initial costs, prices, and economic parameters assumed for various system inputs and outputs for the analyses of manure handling and tillage options on a representative 100-cow dairy farm in Pennsylvania.

Parameter	Value[a]	Structure or Equipment	Initial Cost
Labor wage rate	$12.00 h^{-1}	Tie-stall barn and milking equipment	$360,000
Diesel fuel price	$0.60 L^{-1}	Parlor and free-stall barn	$313,000
Electricity price	$0.10 kWh^{-1}	Gutter cleaner	$27,000
Mailbox milk price	$0.32 kg^{-1}	Skid-steer loader	$23,800
Nitrogen fertilizer price	$0.95 kg^{-1} N	Manure collection pad	$3,000
Corn grain price	$132 t^{-1} DM	Concrete tank storage	$62,000
Soybean meal price	$265 t^{-1} DM	Lined earthen storage	$41,000
Protein feed mix price	$385 t^{-1} DM	Broadcast box spreader	$13,200
Real interest rate	6.0% year^{-1}	Broadcast tank spreader	$19,800
Seed and chemical annual costs:		Injection manure spreader	$27,200
Forage establishment	$270 ha^{-1}	Irrigation equipment	$24,000
Corn	$185 ha^{-1}	Moldboard plow	$17,000
Additional for corn following corn	$37 ha^{-1}	Chisel plow	$10,000
Oats	$74 ha^{-1}	Disk harrow	$14,300
Additional for no-till establishment	$25 ha^{-1}	Field cultivator	$9,400
		Row crop planter	$17,000
		Grain drill	$7,200
		No-till row crop planter	$19,000
		No-till drill	$21,000

[a] Prices were set to represent long-term relative prices in current value, which were not necessarily current prices.

sumed initial costs. Initial costs of structures were amortized over 20 years and equipment was amortized over 10 years using a real interest or discount rate of 6% per year. Important prices in the comparison of these systems are also listed in table 4. All prices were held constant across simulated systems and years of weather.

MANURE HANDLING SYSTEMS

Five manure–handling options were simulated, with the first two using daily hauling and spreading and the last three using long–term manure storage. In all options, conservation tillage was used for crop establishment. The first option used a tie–stall barn where manure was collected using gutter scrapers and field-applied on a daily basis using a box spreader. Bedding material was used to provide manure in a semi-solid form. Manure storage was limited to a concrete pad for short–term storage. Field-applied manure remained on the soil surface up to six months prior to incorporation by tillage. The second option used a milking parlor and free–stall barn. Less bedding was used, producing a slurry manure. Manure was collected using a skid steer loader and field–applied on a daily basis using a broadcast tank spreader.

The final three options all used a parlor and free–stall barn with long–term manure storage. Manure slurry was scraped daily, stored up to six months, and applied to fields in April and October. The three strategies used surface spreading, injection, or irrigation application. For surface application, a tank spreader with a splash plate applicator was used where manure remained on the surface for about three days prior to incorporation. With injection, the spreader included tines to insert the manure under the soil surface. The use of injection was assumed to increase implement draft and reduce field capacity of the application operation by 25%. In the fifth option, manure was applied bi-annually using an irrigation system and incorporated by tillage within five days of application. The manure was in a liquid form, and the storage tank was replaced with a larger lined earthen storage.

Phosphorus loss, net return, and other simulated production results for the five options are shown in table 5. The greatest P losses occurred with a tie–stall barn using semisolid manure (13% DM) and a daily haul system. Use of a free–stall barn and slurry manure (8% DM) reduced total P runoff

loss by 7%. Less loss occurred because the more fluid manure infiltrated into the soil more rapidly. Use of long–term storage with field application of manure in the spring and fall gave a similar total P loss as that with daily hauling of the slurry manure. Direct injection of manure provided a 21% reduction in total P loss compared to surface application of slurry manure, whereas irrigation of liquid manure (5% DM) provided little reduction. Compared to surface application of slurry, soluble P loss was reduced 47% with the use of injection and 9% using irrigation of liquid manure.

Averaged over all simulated years, the practice of using six–month storage with injection had less P loss than using daily haul. This decrease was observed because the potential for P loss in runoff increased the longer the applied P remained on the surface. With daily haul and surface application, most of the manure remained on the surface for months before being incorporated. Six–month storage systems, particularly when combined with manure injection, greatly reduced the amount of time manure P remained on the surface. More importantly, the lowest soluble P loss also was observed using manure injection. As compared to insoluble forms, soluble P has more potential for environmental damage through eutrophication if transported to surface waters (Sharpley et al., 1999). As a result, strategies that decrease the loss of soluble P, as well as total P, from a farm are desirable.

Farm profitability was affected most by the type of barn used. The tie–stall barn was relatively inefficient in the use of bedding, labor, and a few other resources associated with animal housing and management. Although this type of system is commonly used on smaller dairy farms throughout the northeast, a new structure of this type would not normally be built. The most profitable system was a free–stall barn with daily manure hauling. Including a six–month manure storage unit increased production costs and reduced annual profitability by $62/cow. This assumed that the producer was bearing the full cost of constructing a concrete tank. Although the use of injection increased manure handling costs, farm profitability was increased a small amount through more efficient use of nitrogen on the farm (Rotz and Oenema, 2006).

Six–month storage with injection (system with lowest P loss) gave the second-highest net return, suggesting that the

Table 5. Effect of manure handling practices on annual nutrient losses, manure handling costs, production costs, and net return of a representative 100-cow dairy farm in southern Pennsylvania.

Production Parameter	Tie-Stall Barn	Free-Stall Barn			
	Daily Haul	Daily Haul	Six Months of Storage		
	Surface Applied	Surface Applied	Surface Applied	Injected	Irrigated
Erosion sediment loss (kg ha^{-1})	2621	2621	2660	2717	2665
Total P loss (kg P ha^{-1})	1.36	1.27	1.30	1.03	1.28
Leached P (kg ha^{-1})	0.02	0.02	0.02	0.02	0.02
Soluble P (kg P ha^{-1})	0.44	0.34	0.34	0.18	0.31
Sediment P (kg P ha^{-1})	0.90	0.91	0.94	0.83	0.95
Total N loss (kg N ha^{-1})	126	123	129	115	136
Manure handling and bedding cost ($/cow)	244	187	238	258	241
Net feed cost ($/cow)	1263	1273	1282	1257	1291
Milking equipment and labor cost ($/cow)	561	488	488	488	488
Animal facilities cost ($/cow)	291	245	245	245	245
Livestock expenses ($/cow)	369	369	369	369	369
Property tax ($/cow)	48	46	48	48	48
Total production cost ($/cow)	2776	2608	2670	2665	2682
Milk and animal income ($/cow)	3211	3211	3211	3211	3211
Net return to management ($/cow)	435	603	541	546	529

use of manure injection might be best for the producer despite the greater initial investment and operating costs. At least, the use of manure injection can provide reductions in P loss (and volatile N loss; Rotz and Oenema, 2006) with little long–term economic impact on the producer.

TILLAGE SYSTEMS

Conventional, conservation, and no–till tillage systems were evaluated on the same representative dairy farm in Pennsylvania. For these three options, manure was handled as slurry, which was stored up to six months and surface applied with a broadcast spreader. Conventional tillage included moldboard plowing, disking, and two passes with a field cultivator followed by the planting operation. Conservation tillage involved a similar sequence of operations, but a chisel plow was used as the primary tillage operation, and one pass of the field cultivator was removed. For no–till, all tillage operations were removed, and the planting equipment was replaced with implements suitable for no–tillage conditions. Annual chemical costs for crop establishment were also increased by $25/ha under no–till to reflect greater use of pesticides. This additional cost represented about $15/ha for a broadcast herbicide application prior to planting and $10/ha for a broadcast insecticide application.

Overall, practices using conventional tillage resulted in greater P loss than those using conservation or no–till systems (table 6). Greater loss was directly related to greater erosion due to greater disruption of the soil surface and full incorporation of surface residue. Use of a chisel plow reduced total P loss by 46%. Sediment P loss was reduced by 56%, but since manure was not incorporated as thoroughly, soluble P loss increased 62%. Use of no–till reduced total P loss by an additional 27% due to 50% less sediment P loss and 35% greater soluble P loss compared to the use of conservation tillage. The differences in net return among the systems were relatively small. Use of conservation tillage provided a $17/cow improvement in net return compared to convention–

al tillage, and use of no–till provided an additional increase of $26/cow over conservation tillage.

Under no-till management, the combination of undisturbed soil and surface-applied manure reduced erosion and subsequent losses of sediment-bound P, reducing the overall P lost from the farm. However, with this system, the soluble P loss increased compared to conventional or conservation tillage. As a result, no-till should be used in combination with practices that reduce the loss of soluble P, such as the use of manure injection. Although the soil disturbance created by an injection device does not fit well with no-till practices, manure application equipment is being developed to provide injection with minimal soil disturbance.

CONCLUSIONS

A process-based model of soil P transformation and movement was developed to simulate field and farm scale losses of soluble and sediment P. Incorporation of this soil P component into a whole-farm simulation model provides a tool for evaluating the long-term effects of farm management on P losses along with N losses and other aspects of farm performance and economics.

The soil P model was evaluated by comparing simulated and observed losses from six small watersheds representing fields in corn production with 0 to 430 kg P ha^{-1} applied in poultry litter. Simulation of these fields over two years of weather gave sediment erosion losses similar to those observed. Total P loss was accurately predicted when annual P applications were 250 kg ha^{-1} or less, but overestimation of P loss occurred at higher rates. Compared to observed data, soluble P loss was underestimated and sediment P loss was overestimated, but these differences were primarily due to a difference in the definitions of soluble and sediment P between the experimental and modeling studies.

Whole-farm simulation of production systems for a representative dairy farm in Pennsylvania indicated that the use of a six-month manure storage period and field application of manure by injection reduced total P losses from the farm by 19% compared to daily surface application, but an increase in production costs reduced annual farm net return by $57/cow. Use of conservation and no-till systems provided approximate reductions in total P losses of up to 57% compared to a conventional moldboard plow tillage system with small increases in farm profitability. The farm simulation model with the new soil P component provides a tool for evaluating on-farm management of P in the context of whole-farm environmental concerns and economic considerations.

Table 6. Effect of tillage system on annual manure handling costs, nutrient losses, production costs, and net return of a representative 100-cow dairy farm in southern Pennsylvania.

Production Parameter	Tillage System[a]		
	Conv.	Cons.	No-till
Erosion sediment loss (kg ha^{-1})	9739	2660	1000
Total P loss (kg P ha^{-1})	2.39	1.30	0.95
Leached P (kg P ha^{-1})	0.02	0.02	0.02
Soluble P (kg P ha^{-1})	0.21	0.34	0.46
Sediment P (kg P ha^{-1})	2.16	0.94	0.47
Total N loss (kg N ha^{-1})	130	129	129
Manure handling and bedding cost ($/cow)	237	238	247
Net feed cost ($/cow)	1300	1282	1247
Milking equipment and labor cost ($/cow)	488	488	488
Animal facilities cost ($/cow)	245	245	245
Livestock expenses ($/cow)	369	369	369
Property tax ($/cow)	48	48	48
Total production cost ($/cow)	2687	2670	2644
Milk and animal income ($/cow)	3211	3211	3211
Net return to management ($/cow)	524	541	567

[a] Conventional tillage (Conv.) included moldboard plowing, disking, and two passes with a field cultivator followed by planting. Conservation tillage (Cons.) used a chisel plow and one less pass of the field cultivator. No-till was only a planting operation using implements suitable for no-tillage conditions.

REFERENCES

Arnold, J. G., R. Srinivasan, R. S. Muttiah, and J. R. Williams. 1998. Large-area hydrologic modeling and assessment: Part I. Model development. *J. American Water Resources Assoc.* 34(1): 73-89.

Dou, Z., K. F. Knowlton, R. A. Kohn, Z. Wu, L. D. Satter, G. Zhang, J. D. Toth, and J. D. Ferguson. 2002. Phosphorus characteristics of dairy feces affected by diet. *J. Environ. Qual.* 31(6): 2058-2065.

Flanagan, D. C., and M. A. Nearing, eds. 1995. USDA Water Erosion Prediction Project (WEPP) hillslope profile and watershed model documentation. NSERL Report No. 10. West Lafayette, Ind.: USDA-ARS National Soil Erosion Research Laboratory.

Harmel, R. D., H. A. Torbert, B. E. Haggard, R. Haney, and M. Dozier. 2004. Water quality impacts of converting to a poultry litter fertilization strategy. *J. Environ. Qual.* 33(6): 2229-2242.

Jones, C. A., A. N. Sharpley, and J. R. Williams. 1984. A simplified soil and plant phosphorus model: III. Testing. *SSSA J.* 48(4): 810-813.

Kleinman, P. J. A., and A. N. Sharpley. 2003. Effect of broadcast manure on runoff phosphorus concentrations over successive rainfall events. *J. Environ. Qual.* 32(3): 1072-1081.

Neitsch, S. L., J. G. Arnold, J. R. Kiniry, J. R. Williams, and K. W. King. 2002. Soil and water assessment tool: Theoretical documentation – Version 2000. College Station, Tex.: Texas Water Resources Institute.

Renard, K. G., G. R. Foster, G. A. Weesies, D. K. McCool, and D. C. Yoder. 1996. Predicting soil erosion by water: A guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). Agric. Handbook No. 282. Washington, D.C.: USDA.

Rotz, C. A., and C. U. Coiner. 2006. Integrated Farm System Model, Reference Manual. University Park, Pa.: USDA-ARS Pasture Systems and Watershed Management Research Unit. Available at: www.ars.usda.gov/naa/pswmru. Accessed September 2006.

Rotz, C. A., and J. Oenema. 2006. Predicting management effects on ammonia emissions from dairy and beef farms. *Trans. ASABE* 49(4): 1139-1149.

Sharpley, A. N. 1985. The selective erosion of plant nutrients in runoff. *SSSA J.* 49(6): 1527-1534.

Sharpley, A. N., C. A. Jones, C. Gray, and C. V. Cole. 1984. A simplified soil and plant phosphorus model: II: Prediction of labile, organic, and sorbed phosphorus. *SSSA J.* 48(4): 805-809.

Sharpley, A. N., T. Daniel, T. Sims, J. Lemunyon, R. Stevens, and R. Parry. 1999. Agricultural phosphorus and eutrophication. ARS-149. Springfield, Va.: USDA National Technical Center.

U.S. EPA. 2003a. National Section 303(d) List Fact Sheet. Washington, D.C.: U.S. Environmental Protection Agency. Available at: oaspub.epa.gov/waters/national_rept.control. Accessed November 2005.

U.S. EPA. 2003b. 2000 National Water Quality Inventory. Washington, D.C.: U.S. Environmental Protection Agency. Available at: www.epa.gov/305b/2000report/. Accessed November 2005.

Vadas, P. A. 2001. Modeling phosphorus export from agricultural fields in the mid-Atlantic coastal plain: The FHANTM-MACP model. PhD diss. Newark, Del.: University of Delaware.

Vadas, P. A. 2006. Distribution of phosphorus in manure slurry and its infiltration after application to soils. *J. Environ. Qual.* 35(2): 542-547.

Vadas, P. A., P. J. A. Kleinman, and A. N. Sharpley. 2004. A simple method to predict dissolved phosphorus in runoff from surface-applied manures. *J. Environ. Qual.* 33(2): 749-756.

Vadas, P. A., P. J. A. Kleinman, A. N. Sharpley, and B. L. Turner. 2005. Relating soil phosphorus to dissolved phosphorus in runoff: A single extraction coefficient for water quality modeling. *J. Environ. Qual.* 34(2): 572-580.

Vadas, P. A., T. Krogstad, and A. N. Sharpley. 2006. Modeling phosphorus transfer between labile and non-labile soil pools: Updating the EPIC model. *SSSA J.* 70(3): 736-743.

Vadas, P. A., W. J. Gburek, A. N. Sharpley, P. J. A. Kleinman, P. A. Moore, M. L. Cabrera, and R. D. Harmel. 2007. A model for phosphorus transformation and runoff loss for surface-applied manures. *J. Environ. Qual.* 36(2): 324-332.

Williams, J. R. 1995. Chapter 25: The EPIC model. In *Computer Models of Watershed Hydrology*, 909-1000. V. P. Singh, ed. Highlands Ranch, Colo.: Water Resources Publications.